

Ground-dwelling reptile assemblages in selectively harvested dry sclerophyll forest in south-east Queensland

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ABSTRACT

Reptile assemblages were examined retrospectively in relation to five age categories of regrowth since the last selective tree harvest (i.e., 0-10 years, 11-20 years, 21-40 years, 41-50 years and >50 years/virgin). The aim was to identify indicator species that showed consistent response to post-harvest forest regrowth. A total of 47 reptile species were recorded from 95 systematically surveyed sites. Species richness was lower in 11-20 years old regrowth with no effect observed in regrowth aged 40-50 years. This trend was not significant. Reptiles overall, and a subset of 'common' reptiles, were significantly less abundant in 11-20 year regrowth compared to the 0-10 years regrowth. Abundance of reptiles was not significantly higher in older forests (i.e., 41-50 years since harvest and >50 years or virgin sites) than in forests within the 11-40 years since harvest period. The trends in abundance observed amongst all reptiles and common reptiles were largely due to the numbers of skinks. In general, skink abundance was significantly lower in the 11-20 years since harvest period and highest in the first ten years after harvesting. A majority of these skinks belonged to a guild of bark-, log- and rock-dwellers. High shrub density in the 11-20 year old regrowth provides some explanation for lower numbers of skinks recorded. Shrub density may affect the availability of basking sites for reptiles at ground level and the accessibility and searching behaviour of observers. Ambient temperature at the time of survey had a significant effect on the abundance of some skink species, which meant that an explanation was confounded. The common "bark-, log- and rock-dwelling" skinks, *Lampropholis* spp. and *Carlia* spp. demonstrated measurable changes to measurable habitat conditions and disturbances, which are traits of model "indicator" species. The limitations of using common skinks as indicator species are acknowledged and a caution is issued in respect to inappropriate use of the species indicator tool.

Key words: reptiles, indicator species, timber harvesting, guilds.

Introduction

The notion that a species whose ecological traits reflect attributes of sustainability is appealing to forest managers charged with achieving ecologically sustainable forest management. This has given rise to the idea of management "indicator" species (Thomas 1972; Landres *et al.* 1988), recently reviewed by Lindenmayer *et al.* (2000) in an Australian context. The concept is based on a number of precepts that include: species associations exist within nature and therefore some species might act as surrogates for the presence or absence of other species; species are sensitive to environmental change or changes to cyclical patterns outside the norm and; species are sensitive to the effects of a disturbance regime or management designed to mitigate the impacts of disturbance.

A collaborative project between management agencies from Queensland, New South Wales, Victoria, Tasmania and the CSIRO was initiated to identify and combine (where possible) datasets, and share information that could potentially be used to select "indicator" species that reflect some response to management and the presence or absence of other species or guilds of species. It was envisaged that this project would assist in the

identification of sets of species or functional groups that could be monitored efficiently and practically by forest managers. The species, or groups of species, could be expected to provide early indications of change in status for a wider range of species resulting from forest management activities, with particular focus on the effects of forest harvesting.

To address these goals we undertook a retrospective analysis of species response to post-harvest regrowth using data on reptile assemblages from commercial hardwood forests of the south-east Queensland (SEQ) biogeographic region. Timber harvesting in the eucalypt forests of this region has been a low-key operation compared with clear-fall operations that are routine in the southern states of Australia. Since European settlement, early in the 19th century, Queensland's drier forests have been lightly harvested for hewn, split and round timber. As native softwoods and rainforest timbers began to decline in the region, eucalypt forests became the focus for sawlog production (Florence 1996). During the 1950s, a program of silvicultural research, forest inventory, control of harvesting operations through tree-

marking, log measurement and the development of an allocation system was implemented and led to changed harvesting and treatment regimes. Harvesting and silvicultural treatment of stems to encourage productive forests has therefore varied in terms of size classes targeted and extent, respectively since the late 1930's. Stems have been cut as mill logs, hardwood girders, hardwood poles and piles.

While studies in Queensland have recently examined the response of various fauna groups to forest management practices, such as harvesting and burning (Hannah *et al.* 1997; Wormington 2002; Smyth *et al.* 2002), there are no studies that acknowledge impacts of harvesting on reptile assemblages. Elsewhere in Australia a number of studies have examined reptiles in relation to forest harvesting (e.g., New South Wales: Dunning 1986; Lunney and Barker 1986; Lunney *et al.* 1991; Webb 1995; Goldingay *et al.* 1996; Kavanagh and Webb 1998, Victoria: Brown and Nelson 1993; Kutt 1993, Tasmania: Duncan 1995 and Western Australia: Thomson *et al.* 1994-95).

This paper details the results of an investigation into the diversity and abundance of the reptiles in five regrowth age-classes of forest recovering from selective harvesting. Each regrowth age-class is a combination of "time-since-harvest" effects and a harvesting-treatment operation. Regrowth age-classes are not strictly a "chronosequence" representing post-successional states following a single type of harvest/treatment operation but we assumed, for the purposes of our analysis, that harvesting in general increases dead timber and creates greater ground-level exposure to solar radiation. Therefore, post-harvest regeneration of non-commercial flora was likely to affect reptiles no matter which of the four major harvest/treatment regimes had been applied by forestry in the drier forest of south-east Queensland. Species patterns, identified by statistical analyses, have been proposed as potential indicator species that demonstrate responses to harvest management regimes.

Study Area

The SEQ bioregion covers approximately 66,000 square kilometres, extending from Gladstone and Koorombit Tops in the north to the New South Wales border in the south (Figure 1; Sattler and Williams 1999). Major landscape features are a coastal plain of varying width, hills and ranges, three major drainage basins, and coastal mainland and island sand masses. Forests include subtropical rainforests, dry vine forest thickets, tall open forests on more fertile substrates (that include blackbutt *Eucalyptus pilularis*, tallowwood *E. microcorys*, flooded gum *E. grandis* and grey gum *E. propinqua*) and extensive eucalypt forests and woodlands in drier areas (that contain narrow leaf ironbark *E. creba*, silver leaf ironbark *E. melanophloia*, forest red gum *E. tereticornis* and spotted gum *Cormybia citriodora*). Paperbark and wallum heaths occur primarily on less fertile lowlands (Specht and Specht 1999; Young and Dillewaard 1999). The mean annual rainfall ranges from >1500 mm on the coastal ranges to 800 mm in the sub-humid interior (Young and Dillewaard 1999).

Methods

Vertebrate fauna (excluding fish) were surveyed at 249 sites between February and December 1997 for the Comprehensive Regional Assessment (CRA) of the SEQ bioregion. Reptiles were surveyed using active searches for half an hour by day and a half an hour at night by two skilled observers searching an area of 0.5 ha (Eyre *et al.* 1998). Pit-traps, consisting of five buckets installed in the ground at six metre intervals along a 30 metre length of drift fence, were also deployed at some sites. Habitat assessments were conducted at each systematic site according to methods outlined by Eyre *et al.* (1998) and included measures of small trees, large trees, stumps, hollows, foliar cover, ground cover, litter cover, litter depth, logs, rocks, vines, moss, ferns, weeds, grasses and shrubs (Table 1).

The retrospective analyses undertaken in this paper focus only on the dry sclerophyll sites that were systematically surveyed using diurnal and nocturnal searches and pit-traps. Sites were assigned to a regrowth category based on habitat information collected at each site and information contained within the Area Information System (AIS) administered by the Queensland Department of Primary Industries, Forestry (DPI-F). It was not possible to assign some sites to a regrowth category on account of insufficient information. 95 sites (38% of the 249 sites), spanning 29 State Forests across the SEQ bioregion (Figure 1) were allocated to a regrowth category and analysed for this study. In addition to site information collected during the

Table 1. List of habitat and landscape variables measured at each site

Variables	
Landscape	Habitat Structure
Cloud Cover Index AM	Number of small trees (10-65 cm dbh)
Cloud Cover Index PM	Number of large trees (>66 cm dbh)
Precipitation Index AM	Number of stags
Precipitation Index PM	Number of stumps
Maximum Temperature (°C)	Number of hollows (>10 cm)
Minimum Temperature (°C)	Percent Foilage Cover
Total annual rainfall (mm)	
Flat solar radiation	
Wet Topographic Index	
Mean annual increment of tree growth (20 yrs)	
Mean annual increment of tree growth (40 yrs)	
Microhabitat	
Bare ground (%)	Ferns (%)
Fine litter (<6cm diameter, %)	Weeds (%)
Coarse litter (>6cm diameter, %)	Grasses/Herbs (%)
Logs (>10cm diameter, %)	Small shrubs (%)
Rock (%)	Mean litter depth (cm)
Vines (%)	Estimated average log diameter (cm)
Moss (%)	Average logs with hollows(#)

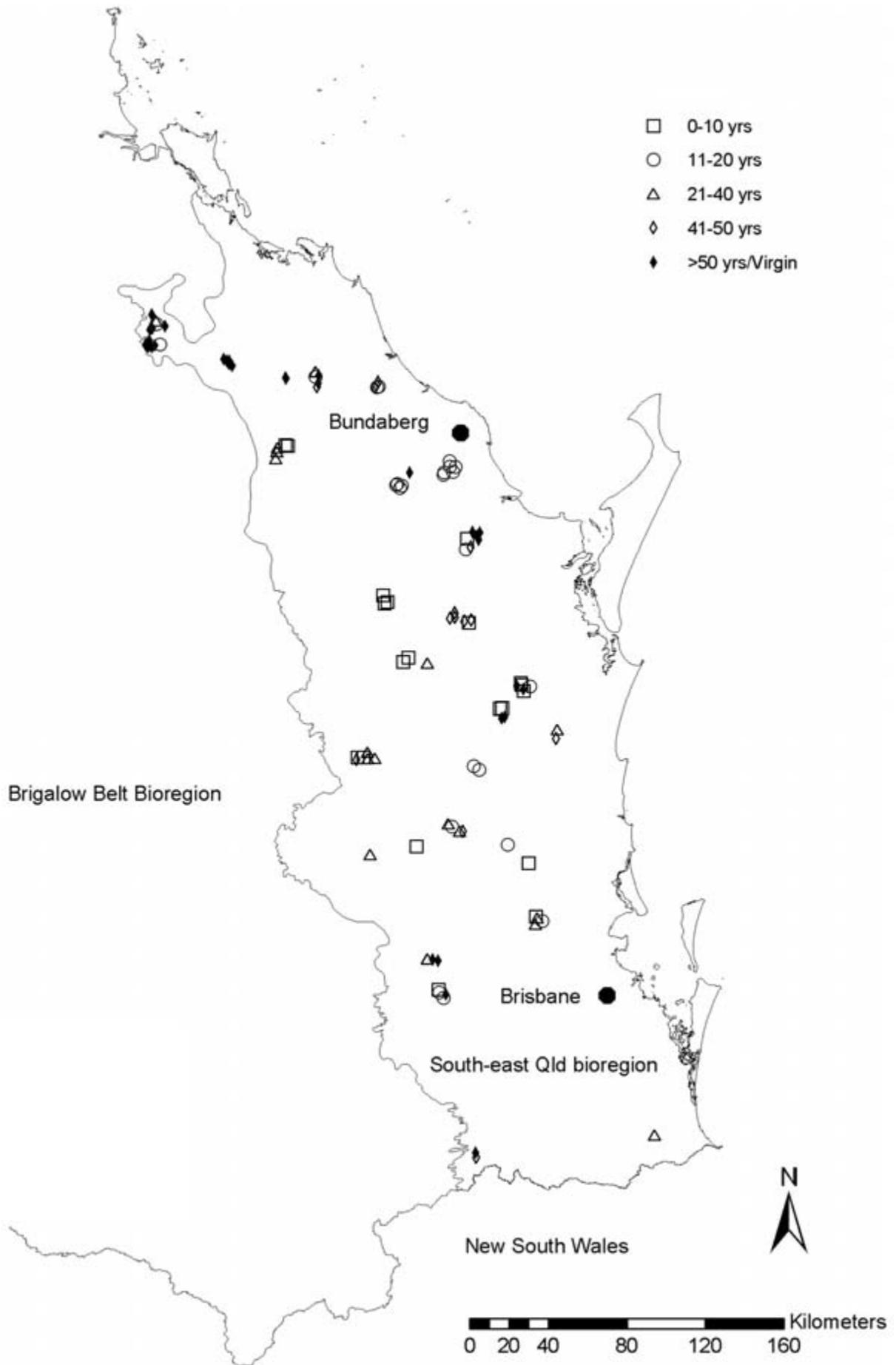


Figure 1. Distribution of sites within South-east Queensland by regrowth category.

survey periods, other variables derived from modelled data were assigned to sites (Table 1). These modelled variables were a productivity index (Mean Annual Increment of tree growth, MAI), total annual rainfall (mm), flat solar radiation and a wet topographic index (Buck et al. 1999).

Data Analyses

Sites were grouped initially into 10-year regrowth categories, but owing to small sample sizes in the 21-30 and 31-40 year categories, the data from these categories

were pooled. The number of sites allocated to each category was: 0-10 years (18 sites), 11-20 years (21 sites), 21-40 years (14 sites), 41-50 years (13 sites) and >50 years or virgin forests (29 sites). Due to funding and CRA reporting timeframes, most reptile surveys were carried out in winter (June to August, 77% of sites), yet they differed across the regrowth groups as follows: 0-10 years (94% winter sites), 11-20 years (86% winter sites), 21-40 years (64% winter sites), 41-50 years (54% winter sites) and the >50 years or virgin forests (76% winter sites).

Table 2. List of reptile species within each functional guild based on sheltering preferences.

Reptile Species	Burrows	Bark & Logs	Rocks	Standing Trees Bark	Standing Trees Hollows	Litter	Camouflage	Water	Low Vegetation	Crevices
SPECIES THAT USE BARK AND LOGS										
<i>Diplodactylus vittatus</i>		✓	✓							
<i>Eulamprus brachysoma</i>		✓	✓							
<i>Morethia taeniopleura</i>		✓	✓							
<i>Ramphotyphlops nigrescens</i>		✓	✓							
<i>Heteronotia binoei</i>		✓	✓	✓						
<i>Underwoodisaurus milii</i>		✓	✓							✓
<i>Anomalopus verreauxii</i>		✓	✓			✓				
<i>Calyptotis lepidorostrum</i>		✓	✓			✓				
<i>Calyptotis scutirostrum</i>		✓	✓			✓				
<i>Calyptotis temporalis</i>		✓	✓			✓				
<i>Eroticoscincus graciloides</i>		✓	✓			✓				
<i>Glaphyromorphus punctulatus</i>		✓	✓			✓				
<i>Lampropholis adonis</i>		✓	✓			✓				
<i>Lampropholis amicula</i>		✓	✓			✓				
<i>Lampropholis delicata</i>		✓	✓			✓				
<i>Lerista fragilis</i>		✓	✓			✓				
<i>Carlia foliorum</i>		✓	✓			✓				
<i>Menetia timlowi</i>		✓	✓			✓				
<i>Saiphos equalis</i>		✓	✓			✓				
<i>Lialis burtonis</i>		✓	✓			✓			✓	
<i>Carlia pectoralis</i>		✓	✓			✓			✓	
<i>Carlia schmeltzii</i>		✓	✓			✓			✓	
<i>Carlia vivax</i>		✓	✓			✓			✓	
<i>Ctenotus arcanus</i>		✓	✓			✓			✓	
<i>Ctenotus robustus</i>		✓	✓			✓			✓	
<i>Ctenotus taeniolatus</i>		✓	✓			✓			✓	
<i>Furina diadema</i>		✓	✓			✓			✓	
<i>Hemiaspis signata</i>		✓	✓			✓			✓	
<i>Demansia psammophis</i>		✓	✓			✓	✓		✓	
<i>Eulamprus quoyii</i>		✓				✓		✓		
SPECIES THAT USE BARK/LOGS, STANDING TREES - HOLLOW AND BARK										
<i>Rhinoplocephalus nigrescens</i>		✓	✓	✓	✓	✓	✓		✓	✓
<i>Morelia spilota</i>		✓		✓		✓	✓		✓	✓
<i>Gehyra dubia</i>		✓	✓	✓	✓					
<i>Oedura rhombifer</i>		✓	✓	✓	✓					
<i>Oedura robusta</i>		✓	✓	✓	✓					
<i>Oedura tryoni</i>		✓	✓	✓	✓					
<i>Eulamprus martini</i>		✓	✓	✓	✓					
<i>Varanus tristis</i>		✓		✓	✓					
<i>Cryptoblepharus virgatus</i>		✓		✓	✓					
<i>Eulamprus tenuis</i>		✓		✓	✓					
SPECIES THAT USE CAMOFLAUGE										
<i>Pogona barbata</i>							✓		✓	
<i>Amphibolurus nobbi</i>			✓				✓		✓	✓
<i>Diporiphora australis</i>							✓		✓	✓
<i>Physignathus lesueurii</i>							✓	✓	✓	
<i>Boiga irregularis</i>					✓		✓			✓
SPECIES THAT USE BURROWS, BARKS AND LOGS, OPEN AIR/CAMOFLAUGE										
<i>Varanus varius</i>	✓	✓			✓		✓			
<i>Varanus gouldii</i>	✓	✓					✓		✓	

Diversity indices (species richness, Shannon-Weiner index H' and Pielou's evenness J ; Zar 1996) and abundances for common reptiles (species with 5 or more records), uncommon reptiles (species with 4 or less records), functional groups (based on sheltering preferences), family groups and individual species were calculated for each site. The functional groups of reptiles were identified according to their preferred shelter. These groups were combinations of burrow-dwellers; bark- and log-dwellers; rock-dwellers; species that utilise the bark on standing trees; species that utilise hollows on standing trees; litter-dwellers; species that use camouflage and/or no cover; aquatic species; low vegetation-dwellers; and species that use crevices (Table 2).

We undertook two analytical procedures in order to examine diversity indices, abundance measures and the species composition of sites in relation to regrowth age and environmental variables: (1) two- and one-factor analysis of variance (ANOVA), and Tukey's multiple comparison testing, to test for differences in diversity and abundance (total and for individual species), and habitat variables among regrowth categories; (2) classification and ordination of sites based on species to determine similarity between the sites in the five regrowth categories. These analytical procedures are elaborated below:

(1) Diversity, abundance and habitat variables across regrowth categories

Diversity and abundance measures were analysed with two-factor analysis of variance using a GLM procedure for unbalanced data in the SAS statistical package (SAS Institute Inc. 2000). The factors in the design were regrowth age (five levels), season (two levels – winter and summer) and the interaction between regrowth age and season. Differences were considered significant at $P < 0.05$ and the least-square means for each category were compared with Tukey's multiple comparison tests. If the interaction between the two factors was not significant, then the data were re-analysed without an interaction term using the GLM procedure. With significant regrowth age and season interactions, separate analyses using one-factor ANOVAs were performed on the diversity and abundance data for each season. Habitat structure variables were also compared across the regrowth categories using the same analytical procedures. A square root transformation of the abundance and diversity data was applied before analysis to improve normality and homoscedasticity. All means and standard errors displayed within graphs are arithmetic and untransformed.

(2) Classification and ordination of survey sites based on species constituency

Multivariate analyses were performed at a species level to determine similarity between the sites in the five regrowth categories. Two-dimensional ordinations were produced using non-metric, multi-dimensional scaling with the PATN package (Belbin 1995). The Bray-Curtis Similarity Matrix was used as the distance measure between sites in the ordination plot; this distance measure is used widely to determine the similarity of sites based on species compositions (Clarke 1993). The association

ratio-ordinal cut level was changed to 0.7 to minimise stress levels and the maximum number of random starts was utilised to stabilise the stress levels. Prior to analysis, the data were checked for outliers and transformed to correct for non-normality and heteroscedasticity. Square root transformation produced the greatest normalising effect on the data.

The ordination patterns of the sites were correlated with the individual reptile species using principal (axis) correlation coefficients (PCC) (Belbin 1995). The correlation coefficients for each species association were tested for significance using a Monte-Carlo randomisation technique (MCAO) (Belbin 1995). A one-way analysis of similarity (ANOSIM) analysed the significant differences in the composition of the reptile community between each regrowth category (Clarke and Green 1988). With each pair-wise comparison, Bonferroni corrections (critical value of 0.05) were applied to the test statistic to make an allowance for multiple comparisons across each regrowth category.

A comparison of individual reptile species and local habitat variables were investigated using this ordination. Sites were spaced in the ordination in relation to the composition of common reptiles and habitat variables were overlaid as extrinsic factors using PCC and MCAO. The vectors of the significant habitat variables were directed towards sites with associations.

Results

Analyses were performed on 47 reptile species (58% of the reptiles species recorded during the CRA), comprising 20 common (Table 3) and 27 uncommon species (Table 4). The most commonly recorded species were the skinks *Carlia pectoralis*, *Cryptoblepharus virgatus*, *Carlia foliorum*, *Anomalopus verreauxii* and the gecko *Heteronotia binoei*.

Reptile diversity, abundance and habitat features across regrowth categories

Diversity: Overall the mean species richness, diversity and evenness of reptile species did not differ significantly with regrowth age. A trend ($F = 1.98$, $p = 0.10$) in the mean species richness of common reptiles indicated fewer species were recorded in the 11-20 year category compared to the 41-50 year category (Figure 2).

Total reptile abundance: The mean abundance of all reptiles was significantly affected by regrowth age ($F = 3.13$, $p = 0.02$). Figure 3A shows the mean abundance of all reptiles pooled within sites across regrowth categories. Significantly more reptiles were recorded in the 0-10 year old regrowth than in that aged 11-20 and 21-40 years. The mean abundance of pooled common reptiles within sites also differed between regrowth categories ($F = 2.88$, $p = 0.03$, Figure 3b), with a greater mean abundance in the 0-10 year category compared to the 11-20 year category. The mean abundance of uncommon reptiles did not differ significantly with regrowth age.

Table 3. List of common forest-dwelling reptile species recorded from 95 sites analysed for the WAPIS project from the original 249 standard sites surveyed in the south-east Queensland biogeographic zone during the Comprehensive Regional Assessment.

Family	Species	Total No of Records	Percentage of Sites (n=95)
Agamidae	<i>Diporiphora australis</i>	6	5
Elapidae	<i>Rhinoplocephalus nigrescens</i>	17	16
Gekkonidae	<i>Diplodactylus vittatus</i>	17	13
	<i>Gehyra dubia</i>	19	19
	<i>Heteronotia binoei</i>	59	19
	<i>Oedura rhombifer</i>	24	14
	<i>Oedura tryoni</i>	35	22
Scincidae	<i>Anomalopus verreauxii</i>	50	13
	<i>Calyptotis scutirostrum</i>	28	12
	<i>Carlia pectoralis</i>	187	51
	<i>Carlia vivax</i>	20	12
	<i>Cryptoblepharus virgatus</i>	154	58
	<i>Ctenotus arcanus</i>	19	7
	<i>Ctenotus taeniolatus</i>	13	8
	<i>Eulamprus martini</i>	34	16
	<i>Lampropholis amicula</i>	27	14
	<i>Lampropholis delicata</i>	21	9
	<i>Lerista fragilis</i>	23	13
	<i>Carlia foliorum</i>	105	44
	<i>Morethia taeniopleura</i>	11	7

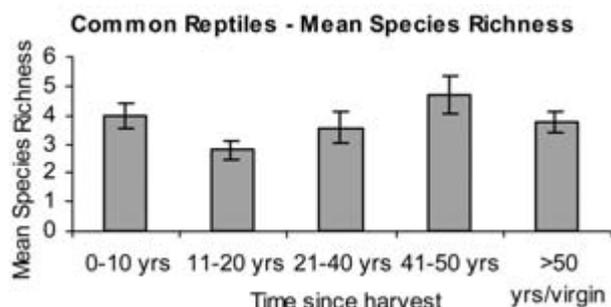


Figure 2. Differences in the mean species richness of common reptile species. Mean species richness refers to the mean number of reptiles recorded per site; all presented means and standard errors are untransformed. Sample sizes same across figures 2-7.

Functional group abundance: All but one of the functional groups showed no significant difference in mean abundance across the regrowth categories. The abundance of the combined “bark-, log- and rock-dwelling” species differed significantly between the regrowth categories ($F=2.91$, $p=0.03$, Figure 4). Tukey’s multiple comparison tests did not separate out any of the means, however the mean abundance of individuals in the 21-40 year category was significantly less than the 0-10 ($p=0.09$) and >50/virgin ($p=0.09$) categories at the 0.1 level of significance. Abundance in the 11-20 year category was similar to that of the 21-40 year category, but did not approach significance.

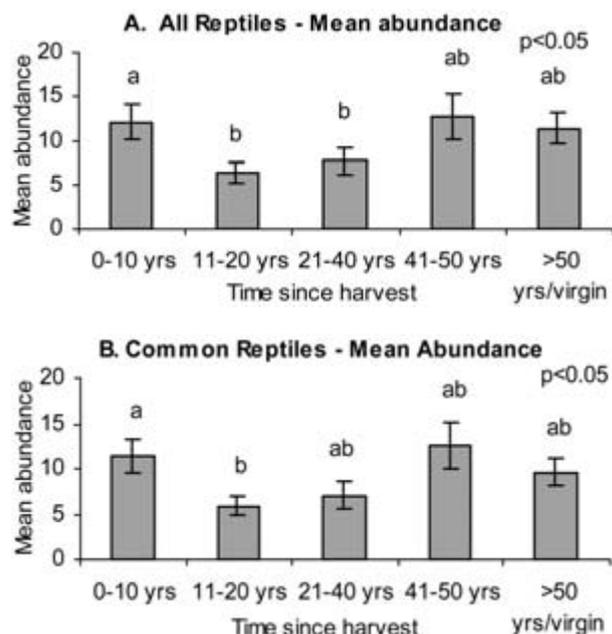
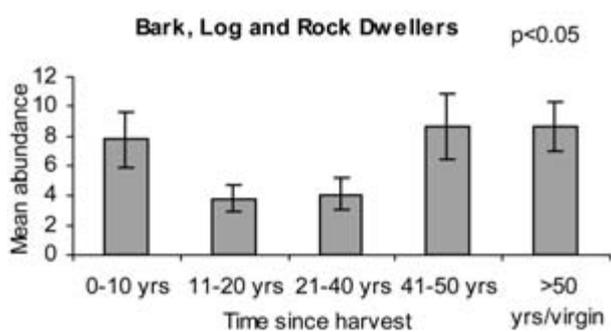


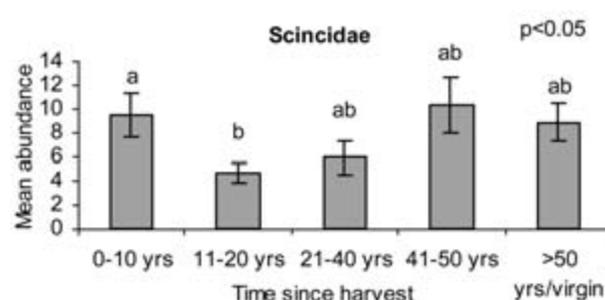
Figure 3. Differences in the mean abundance of A. all reptile species and B. common reptile species. Mean abundance refers to the mean number of reptiles recorded per site; means and standard errors presented here are untransformed. Letters are the results of the Tukey multiple comparison test on means, so means with the same letters are not significantly different. Sample sizes same across figures 2-7.

Table 4. List of uncommon forest-dwelling reptile species recorded from 95 sites from the original 249 standard sites surveyed in the south-east Queensland bio-geographic zone during the Comprehensive Regional Assessment.

Family	Species	Total No of Records	Percentage of Sites (n=95)
Agamidae	<i>Amphibolurus nobbi</i>	2	2
	<i>Physignathus lesueurii</i>	5	2
	<i>Pogona barbata</i>	1	1
Boidae	<i>Morelia spilota</i>	1	1
Colubridae	<i>Boiga irregularis</i>	1	1
Elapidae	<i>Demansia psammophis</i>	3	3
	<i>Furina diadema</i>	1	1
	<i>Hemiaspis signata</i>	1	1
Gekkonidae	<i>Oedura robusta</i>	9	4
	<i>Underwoodisaurus milii</i>	2	2
Pygodidae	<i>Lialis burtonis</i>	2	2
Scincidae	<i>Calyptotis lepidorostrum</i>	1	1
	<i>Calyptotis temporalis</i>	15	1
	<i>Carlia schmeltzii</i>	3	3
	<i>Ctenotus robustus</i>	3	1
	<i>Eroticoscincus graciloides</i>	1	1
	<i>Eulamprus brachysoma</i>	2	2
	<i>Eulamprus quoyii</i>	6	2
	<i>Eulamprus tenuis</i>	2	2
	<i>Glaphyromorphus punctulatus</i>	2	1
	<i>Lampropholis adonis</i>	13	2
	<i>Menetia timlowi</i>	2	2
	<i>Saiphos equalis</i>	2	1
	Typhlopidae	<i>Ramphotyphlops nigrescens</i>	1
Varanidae	<i>Varanus gouldii</i>	1	1
	<i>Varanus tristis</i>	1	1
	<i>Varanus varius</i>	1	1


Figure 4. Differences in the mean abundances of bark-, log- and rock-dwellers. Mean abundance refers to the mean number of reptiles recorded per site; all presented means and standard error are untransformed. Sample sizes same across figures 2-7. (n=18 for 0-10yrs; n=21 for 11-20yrs; n=14 for 21-40yrs; n=13 for 41-50yrs and n=29 for >50yrs/virgin)

Family group abundance: The mean abundance of skinks (Family Scincidae) across the regrowth categories was significant ($F=2.69$, $p=0.04$, Figure 5). Mean skink abundances were significantly lower in 11-20 year-old regrowth when compared to that aged 0-10 years.


Figure 5. Differences in the mean abundance of Scincidae. Mean abundance refers to the mean number of skinks recorded per site; all means and standard error are untransformed. Letters are the results of the Tukey multiple comparison test on least-square means, so means with the same letters are not significantly different. Sample sizes same across figures 2-7.

Common species abundance: Three skink species were significantly influenced by regrowth age, season or the interaction between season and regrowth age. *Ctenotus arcanus* abundance differed significantly with regrowth age ($F=4.97$, $p<0.01$) and season ($F=8.14$,

$p < 0.01$) and there was evidence of an interaction effect between the two factors ($F = 4.87$, $p < 0.01$). Season strongly influenced the species response to regrowth age as greater numbers were recorded in summer and particularly within the 21-40 and >50/virgin categories (see Figure 6a). Significantly more *C. arcaneus* were recorded in the 21-40 category than in the 11-20 and 41-50 categories. Separate analyses of abundances for summer and winter showed no significant differences between regrowth categories. Overall, this response to harvesting may be attributed to a concentration of the species at two forest sites, Kroombit Tops and Squirrel Creek State Forests, surveyed during summer.

For *Eulamprus martini* there was an overall significant difference in abundance across the regrowth categories ($F = 2.81$, $p = 0.03$). Higher mean numbers were detected in the 11-20 year old regrowth category compared with the 41-50 and >50/virgin categories. There was a significant interaction between season and regrowth age ($F = 4.02$, $p < 0.01$). Analysis of this species in summer ($F = 7.61$, $p < 0.01$, Figure 6b) showed significantly higher numbers occurred in the 11-20 regrowth category than in the 21-40, 41-50 and >50 years/virgin categories. Records for this species were distributed across four State Forests (Squirrel Creek, Bania, Kroombit Tops and Bingera State Forests).

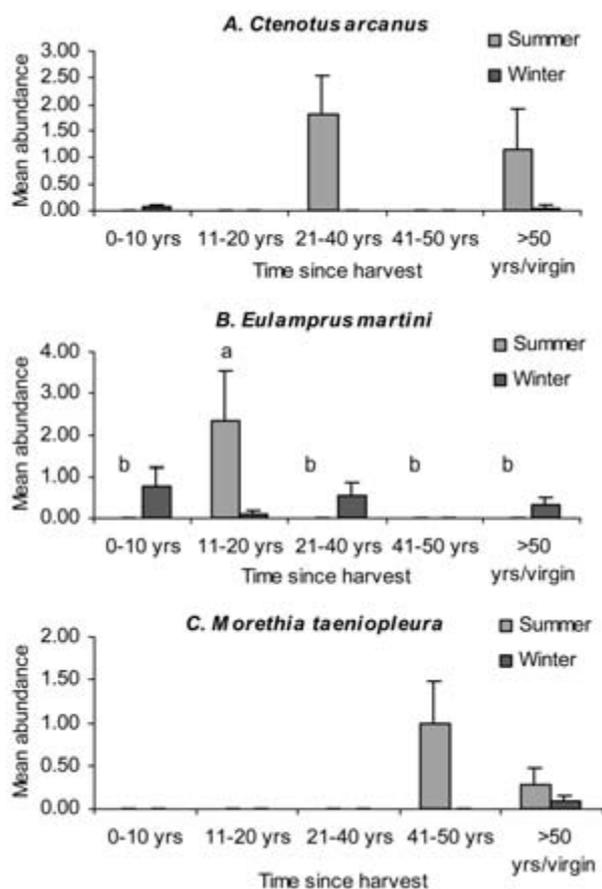


Figure 6. Differences in the mean abundance of *A. Ct. arcaneus*, *B. E. martini*, *C. M. taeniopleura*, with time since last harvest and season. Mean abundance refers to the mean number of individuals recorded per site; all presented means and standard error are untransformed. Sample sizes same across figures 2-7.

There were significant regrowth age ($F = 3.26$, $p = 0.02$), season ($F = 3.96$, $p = 0.05$) and regrowth age by season effects ($F = 3.19$, $p = 0.02$) for *Morethia taeniopleura*. Higher numbers of *M. taeniopleura* were recorded in the 41-50 year old regrowth category compared to the 11-20 and 21-40 year categories. Summer abundances were considerably greater than winter numbers and predominantly higher within the 41-50 years regrowth category (Figure 6c). This species was detected in Glenbar, King, Wongi and Kroombit Tops State Forests.

Habitat comparisons across regrowth categories: Analyses at the regrowth age level, showed that only one habitat variable differed significantly across the categories. The mean percentage of small shrubs was statistically significant ($F = 2.92$, $p < 0.05$, Figure 7), with a higher percentage of shrubs recorded at sites allocated to the 11-20 year harvest category than in the 0-10 years and >50 years/virgin categories.

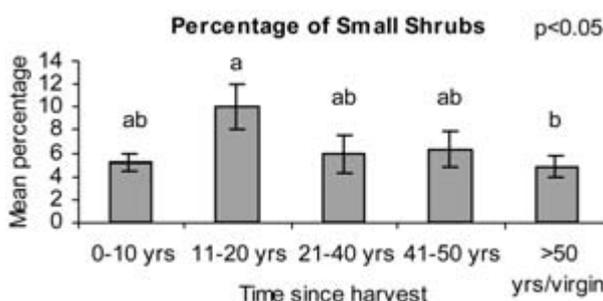


Figure 7. Differences in the mean percentage of small shrubs. Mean percentage refers to the mean number of small shrubs recorded per site; all means and standard error are untransformed. Letters are the results of the Tukey's multiple comparison test on least-square means, so means with the same letters are not significantly different. Sample sizes same across figures 2-7.

Classification and ordination of sites based on species composition

The ordination of sites based on common reptile species showed no clear distinction between the regrowth categories (Figure 8). The ordination had a relatively high stress level of 0.24, suggesting that the pattern in the reptile community varied extensively not only between sites but also within each regrowth category. This is clarified by the comparison, using ANOSIM, which found no significant differences among reptile communities within each regrowth category.

The distributions of several common reptile species were highly correlated with the ordination pattern (vectors are shown in Figure 8). Three skink species and their respective vectors were aligned towards the ordination space with forest sites harvested more than 41 years ago and virgin forest sites. These species were *Carlia pectoralis* ($R = 0.86$, $p < 0.01$), *Morethia taeniopleura* ($R = 0.53$, $p < 0.01$) and *Carlia foliorum* ($R = 0.46$, $p < 0.01$).

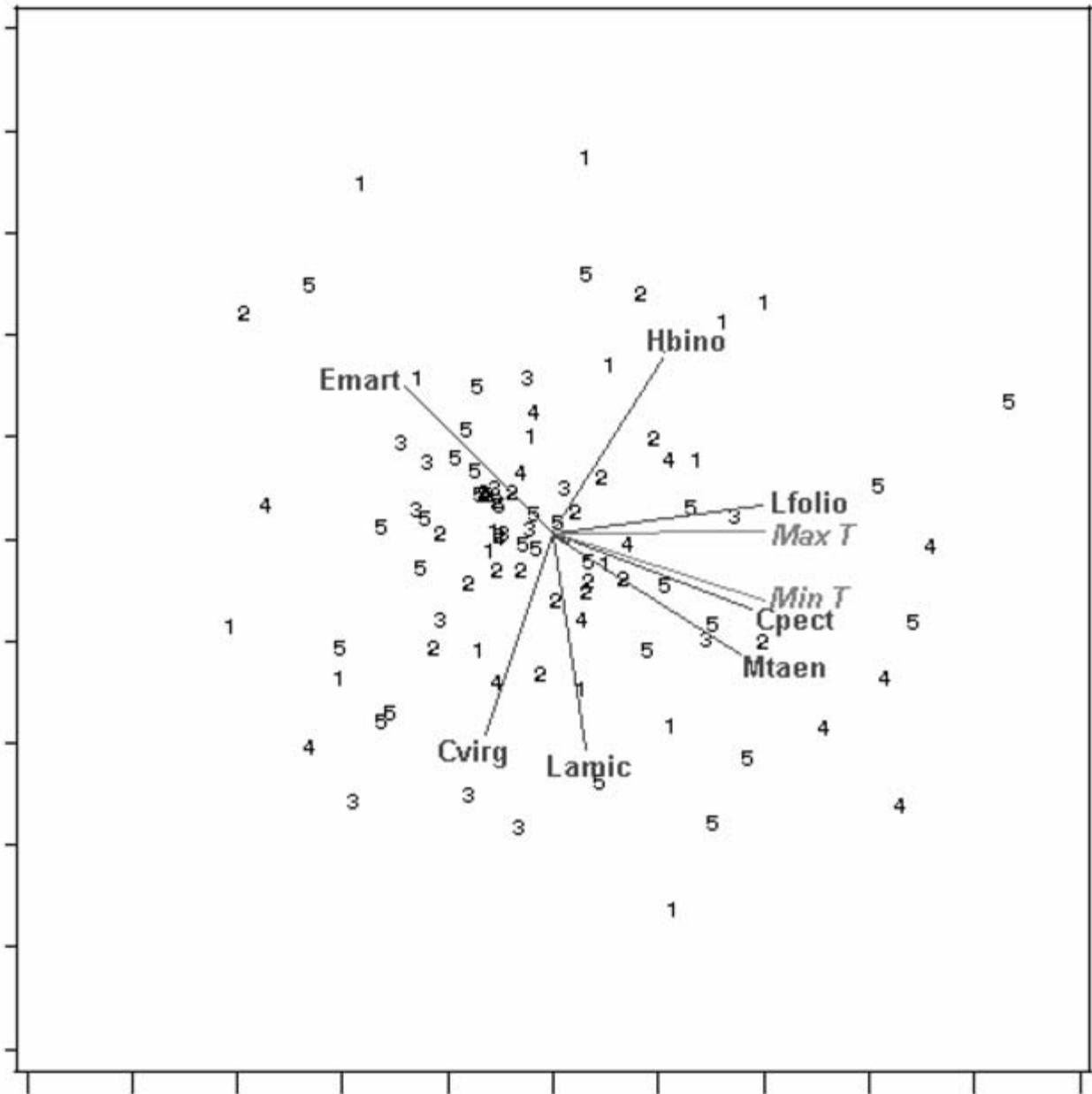


Figure 8. Two-dimensional ordination plot of all sites based on common reptile species composition and abundance. Stress = 0.24, time since harvest categories are 1 = 0-10 years, 2 = 11-20 years, 3 = 21-40 years, 4 = 41-50 years, 5 = greater than 50 years and virgin forests. Vectors of reptile species significantly correlated ($p < 0.01$) with the ordination pattern from the PCC analysis. Lfolio = *Carlia foliorum*; Cpect = *Carlia pectoralis*; Mtaen = *Morethia taeniopleura*; Lamcu = *Lampropholis amicula*; Emart = *Eulamprus martini*; Cvirga = *Cryptoblepharus virgatus* and Hbino = *Heteronotia binoei*. Overlaid environmental variables significantly correlated ($p < 0.05$) with the ordination pattern from the PCC analysis. Min = minimum temperature, Max = maximum temperature; recorded on the day of reptile search.

The abundances of other skink species, *Cryptoblepharus virgatus* ($R=0.72$, $p < 0.01$) and *Lampropholis amicula* ($R=0.46$, $p < 0.01$), were correlated with the ordination pattern. The vectors for these species pointed towards the ordination space in which the majority of forest sites were logged in the last 10 years and over 21 years prior to the CRA survey. The vectors for *Eulamprus martini* ($R=0.49$, $p < 0.01$) and *Heteronotia binoei* ($R=0.43$, $P < 0.01$) were also correlated with the ordination space of forest sites harvested in 0 to 10 years, 11 to 20 years and greater than 50 years regrowth categories.

For all common reptile species, only two habitat characteristics, minimum ($R=0.40$, $p < 0.01$) and maximum ($R=0.38$, $p < 0.01$) daily temperature, were significantly correlated with a majority of sites harvested more than 41 years ago in the ordination pattern. *C. pectoralis*, *M. taeniopleura* and *L. foliorum* were associated with this ordination space, indicating that temperature had a significant effect on their occurrence, mainly on sites in the 41-50 years since harvest category. The proportion of summer sites sampled in this category was higher than any other category.

Discussion

The use of indicator species as surrogates for other aspects of biodiversity or as surrogates for demonstrating changes in ecological processes or ecosystem patterns has come under much scrutiny and, as a management tool, should be treated with some care (Landres *et al.* 1988; Spellerberg 1994; Burgman and Lindenmayer 1998). While being cautious about the future use of indicator species, we also predict that debate over this concept will continue for some time into the future, given the imperatives placed upon the need for such a tool - forest managers need practical measures for assessing change in the dynamics of systems. Whether indicator species ultimately become a useful tool or not, the study outlined in this paper has been worthwhile from two perspectives. Firstly, it permitted us to investigate the appropriateness of using the large, under-utilised CRA dataset for seeking patterns and relationships between species and forest management. Secondly, the dataset has provided a means for seeking relationships between species, environmental variables and management regimes in Queensland's harvested, dry hardwood forests.

Timber harvesting is expected to affect reptile communities in a number of ways, some examples of which include the following: canopy openings created by the removal of trees can lead to greater light and heat penetration to the forest floor and thus provide an increase in available basking area at ground level; increased debris created by harvesting events may provide ground-level refuges not present in naturally-occurring forests; altered microclimates created at ground-level may negatively impact fossorial species; harvesting may simplify forests by decreasing the complexity of vegetation structure and reducing the range and abundance of habitats; harvesting and subsequent post-harvest management, such as top-disposal burning and other such practices, may also lead to vegetative associations that are different floristically and structurally to the naturally-occurring forest (*e.g.* change in species mix, change in density of shrub layers or ground layers). These scenarios identify the need to examine any differences that occur in reptile assemblages between harvested and non-harvested forests and to identify relationships between reptile composition and abundance, and states of post-harvesting forest succession.

Why reptiles as a target for seeking relationships with timber harvesting?

Studies of the impacts of human-induced disturbances on reptile abundance and species composition have included assessments of forest fragmentation (How and Dell 1994; Hadden and Westbrooke 1996; Smith *et al.* 1996; Mac Nally and Brown 2001), grazing (Mathieson *et al.* 1999; Brown 2001), fire (Borsboom 1983; Braithwaite 1987; Tolhurst *et al.* 1992; Trainor and Woinarski 1994; Bamford 1995; Hannah and Smith 1995; Hannah *et al.* 1997; Singh *et al.* 2002) and mining (Twigg and Fox 1991; Letnic and Fox 1997; Taylor and Fox 2001). These studies have indicated that reptiles, as a group, are sensitive to human use and management of forests. Within Queensland, and indeed across the eastern seaboard of Australia, other groups of vertebrates,

particularly hollow-dependent arboreal mammals, have received considerably more attention than reptiles in terms of response to harvesting and it was partly due to this bias that we commenced this study of reptiles.

Why the Comprehensive Regional Assessment (CRA) dataset?

The standardised vertebrate surveys, undertaken during the CRA process, produced a large dataset on species throughout the bioregion. Although surveys were stratified on forest type and not harvesting history, habitat variables indicating where harvesting had occurred, were collected. Until this study, the CRA dataset has not been analysed with regards to establishing relationships between faunal assemblages and harvesting history. To perform this task, the data set was initially sorted to remove records with no information on harvesting history (or where it could not be easily obtained). Sites located within forest types other than dry sclerophyll forests, and additional sites that did not contain a full suite of reptile survey techniques or estimated habitat parameters, were also removed. As a result of the filtering process, the data set was reduced to less than half (38 %) of its original size, reducing the power of statistical tests.

Our investigation of the effects of harvesting on fauna did not deal with harvest intensity, as no real estimates on the harvesting regime were available. Information on the time elapsed since last harvest was accessed from a centralised, DPI-F database, considered to be of variable reliability. Other confounding factors that we assumed were constant across all sites included the fire and grazing management histories. This information was not included in our analyses. In general, the dry forests of SEQ have been subjected to regular fire events for more than 50 years, either as a means of controlling the build up of fuel loads to protect property and investment or by grazing lessees within State Forests to promote fresh grass growth.

In Queensland, there is an inextricable nexus between harvesting, fire and other forest management, such as grazing. It is these relationships that can complicate a retrospective study such as the one reported here. The importance of fire in shaping reptile communities is well documented. Most reptile populations are not fatally affected by the initial impact of fire, being able to take cover in burrows, logs, crevices, water pools and under rocks, or move to new areas where available (Suckling and Macfarlane 1984; Caughley 1985; Tolhurst *et al.* 1992; Friend 1993; Bamford 1995). However, fire alters habitats, such as leaf litter and fallen woody debris, utilised by reptiles for basking, sheltering, oviposition and food supply, making reptiles more susceptible to predation and starvation (Suckling and Macfarlane 1984; Tolhurst *et al.* 1992; Friend 1993). Hence, the abundance of reptiles in the successional stages of forest regeneration after fire is influenced by the availability of the habitats of the various species (Caughley 1985; Friend 1993; Trainor and Woinarski 1994; Hannah *et al.* 1997). The response of individual reptile species will vary considerably with differing fire regimes. Unfortunately, fire records for Queensland's forests are generally poor; we emphasize that we have had to assume that fire regimes were equivalent across the State Forests studied here.

The effects of environmental variables, regrowth age and habitat disturbance on reptiles

Expectations

It is reasonable to suggest that the more heliothermic reptile species would initially increase in abundance within the first 10 years after harvest as harvesting practices open up the canopy, resulting in greater sunlight penetration and availability of substrates for basking and sheltering. As the forest regenerates, the density of the understorey vegetation increases, decreasing the amount of insolation to preferred substrates, disadvantaging some basking reptile species while favouring others. With increasing time, the structure of the vegetation typically becomes more open and creates a greater range of preferred habitats for reptiles, resulting in increased total abundance and a greater diversity of reptiles. While there were significant relationships and trends in reptile richness and abundance across regrowth age categories, clear patterns were probably confounded by the significant temperature effects occurring at survey and the differing proportions of ambient temperatures at survey in each age-class category. We are therefore cautious and somewhat circumspect about the patterns observed in the reptile assemblages among the regrowth categories derived from our analyses. This said some of the relationships do not differ from those previously observed in other studies. We indicate in the following discussion where we believe that associations between species and regrowth age are likely to be confounded by the sampling artefact of temperature effects and unequal sampling effort across summer and winter, between the regrowth categories.

Observation – this study

The reptile communities of SEQ, as measured by species richness, total reptile abundance and common reptile abundance, showed significant responses to the regrowth categories, declining significantly in the period 11-40 years after harvest and then recovering to pre-harvest levels. It appears that one reptile “guild” (bark-, log- and rock-dwellers) and one family group (Scincidae), most of which belong to the aforementioned guild, contributed to the overall patterns in species richness and abundance. We suggest that greater shrub density in 11-20 year old regrowth would provide partial explanation for the suppression in numbers of these species and species groups. A number of reptile species, including *Carlia pectoralis*, *Morethia taeniopleura*, *Carlia foliorum* and *Ctenotus arcanus* were affected by the sampling artefact mentioned previously.

Observation - other studies

Other studies have reported trends in reptile numbers for some species over time since harvesting similar to those recorded in this study, *i.e.* the abundance and diversity of reptiles is lowest in regenerating forests and highest in mature forest and recently harvested forests (Lunney *et al.* 1991; Kutt 1993; Brown and Nelson 1993). The declines in reptile abundance during the regrowth stages of regeneration have been attributed

previously to the increased layering of vegetation and decreasing amounts of insolation reaching preferred substrates (Lunney *et al.* 1991; Brown and Nelson 1993; Kutt 1993; Goldingay *et al.* 1996).

This study, like previous studies, has demonstrated that small skinks respond to harvesting. However, not all species within a group (such as the skinks) respond in the same way over time to harvesting. Lunney *et al.* (1991) examined the effects of various logging operations on three skink species within dry coastal forests of south-eastern Australia. They found that two species not affected immediately by harvesting still declined in abundance in 10 to 15 year old regrowth, whereas the third species, that prefers to inhabit sheltered gullies, increased in abundance in 10 to 15 year old regrowth. *Lampropholis delicata* is one skink species that appears to be adversely affected by harvesting activity. Lunney *et al.* (1991) found that abundances of *L. delicata* were higher in forests selectively harvested over 20 years ago than in recently logged (0-1 years) and 10 to 15 year-old clear-felled regrowth. They suggested that *L. delicata* does not prefer sites with reduced cover, as it exposes individuals to heat stress and predation. Kutt (1993) reported that *L. delicata* occurred more often in unthinned, sheltered 25-35 year old regrowth forest than in thinned 25-35 year old regrowth forest, yet it was not recorded in mature forest. As *L. delicata* prefers moist and sheltered habitats, abundance levels within regenerating forests may be influenced by increasing insolation (Kutt 1993; Cogger 2000).

Conclusion

In general, this and other studies show that reptiles are responsive to forest harvesting. Common skinks of the “bark-, log- and rock-dwelling” guild association, and in particular *Lampropholis* spp. and *Carlia* spp. decline in abundance 20–40 years after harvest, paralleling a general decline in overall reptile abundance and diversity. In effect, these species appear to act as surrogates for the presence or absence of other species, they are sensitive to environmental change and respond to the effects of a disturbance regime. We therefore flag this group of species as demonstrating the traits of “model” indicator species, as previously discussed by Smyth *et al.* (2000), and suggest that they could have an important function in monitoring sustainable forest management. We are still cautious that the limitations of the species indicator tool be recognised and that care be taken when applying it across different climate conditions, particularly for species dependent on ambient temperatures.

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